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FORTY-ONE YEARS OF VEGETATION CHANGE ON PERMANENT TRANSECTS IN NORTHEASTERN CALIFORNIA: IMPLICATIONS FOR WILDLIFE

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We examined changes in ground-cover, tree overstory, and distribution of Great Basin plant species on 69 line-intercept transects measured four times from 1957 to 1998. Tree overstory increased for all years as live shrubs declined and dead shrubs increased since 1967. Increases in the distribution, ground-cover, and overstory of western juniper is an important factor in reductions of understory vegetation and an influence on this region's ecological potential. The distribution of cheatgrass is important to consider when evaluating impacts to native plants and habitats important to wildlife. Trends in shrub-dependent wildlife populations appear to have been affected by changes in shrub cover.

INTRODUCTION

In the Great Basin, changes in native plant communities from the impacts of Euro-American settlement have been well discussed (Wagner 1978, Gruell¹ 1986, Berger and Wehausen 1991). Uncontrolled livestock grazing and disruptions in natural fire cycles are commonly attributed to converting native grass rangelands to the shrub and tree dominated habitats of today (Cottam and Stewart 1940, Shinn 1980, Arno and Gruel 1983, Miller and Rose 1999, Miller et al. 2000). In addition, the introduction and expansion of non-native vegetation has greatly altered ecological relationships on

¹Gruell, G. E. 1986. Post-1900 mule deer irruptions in the intermountain west: Principle cause and influences. United States Department of Agriculture, Forest Service, General Technical Report INT-206, Ogden, Utah, USA.

many Great Basin rangelands (Young et al. 1972, Mack 1981). Although major shifts in habitats have occurred across the Great Basin, direct long-term measurements of these changes are uncommon due to the lack of historical data.

Many factors can effect the dynamics of wildlife populations, but habitats are the primary influence behind changes in distribution and productivity (Caughley 1969, Estes 1996, Livaitis et al. 1999). Changes in Great Basin wildlife populations would be expected from extensive alterations of post-settlement plant communities. Indeed, population trends of some shrub-obligate species have responded to changes in shrub composition (Clements and Young 1997, Connely and Braun 1997), but quantified estimates of long-term changes in vegetation needed to assess these relationships have not been well documented.

Specific knowledge of changes in plant species composition and abundance is important when developing resource management objectives. By identifying temporal and spatial patterns in the composition of habitats, wildlife population trends may be better understood and effects from land management more accurately assessed.

We measured permanent line-intercept transects with standardized survey methods to quantify changes in ground-cover and tree overstory from 1957 to 1998. This study provided the unique opportunity to compare direct measurements of vegetation change with trends in shrub-dependent wildlife populations.

STUDY AREA

The study was conducted on the Devil's Garden in Modoc County, California (latitudes 41°30' to 41°52'; longitudes 121°22' to 120°52'). A subregion of the Modoc Plateau and part of the Great Basin Province (Hickman 1993), the Devil's Garden consists primarily of public-owned lands administered by the Modoc National Forest (MNF). Large-scale effects on habitats occur from timber, fire, and range management programs. Trends in timber harvest have declined on the MNF from annual highs of >100 million board feet (mbf) in the 1970's to <7 mbf in 2001 (USFS, Pacific Southwest Region, unpublished data). Fires are actively suppressed on the Devil's Garden and have significantly altered pre-settlement fire return intervals of <20 years (Miller et al.² 2001), causing increased fuel loads and occasional catastrophic wildfires (USDA³ 1991, Arno⁴ 1996). Prior to formation of the MNF in 1909, an estimated 75,000 cattle and horses and 125,000 sheep were grazing the Devil's Garden region (Interstate Deer Herd

²Miller, R., C. Baisan, J. Rose, and D. Pacioretty. 2001. Pre and post settlement fire regimes in mountain big sagebrush steppe and aspen: the northwest Great Basin. Final report 2001 to the National Interagency Fire Center.

³United States Department of Agriculture. 1991. Modoc National Forest Land and Resource Management Plan, Pacific Southwest Region.

⁴Arno, S. F. 1996. The seminal importance of fire in ecosystem management-impetus for this publication. Pages 3-5 in: C. C. Hardy and S. F. Arno, editors. The use of fire in forest restoration. Gen. Tech. Rep. INT-GTR-341. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

Committee 1951). Currently, the MNF provides 122,500 animal-unit-months which is 23% of permitted livestock forage allocated for California's 18 national forest (USDA³ 1991).

Habitats on the Devil's Garden are generally characterized as juniper savanna and shrub steppe, with some extensive forested areas of montane pine (Kuchler 1977). Plant species associated with these habitats consists primarily of western juniper, *Juniperus occidentalis*; ponderosa pine, *Pinus ponderosa*; bitterbrush, *Purshia tridentata*; big sagebrush, *Artemisia tridentata*; rabbitbrush, *Chrysothamnus* spp.; and mahala mat, *Ceanothus prostratus*; with *Prunus* spp. and curleaf mountain-mahogany, *Cercocarpus ledifolius*, less abundant. Herbs are largely represented by blue grasses, *Poa*; balsamroot, *Balsamorhiza sagittata*; mule's ear, *Wyethia mollis*; buckwheats, *Eriogonum*; and cheatgrass, *Bromus tectorum*, a non-native species accidentally introduced to northeastern California in the late 19th century (Bossard et al. 2000).

Geologically, the Modoc Plateau is a relatively flat transition zone between the Cascade volcanic province to the west and the Great Basin to the east (McDonald⁵ 1966). Elevations range from 1300-1800 m, with 30-40 cm of annual precipitation. A detailed description of this region's unique geology, vegetation, and fauna is provided by Salwasser⁶ (1979).

METHODS

Sixty-nine linear 61-m transects were established prior to 1957 on the 500 km² "key area" of winter range for the Interstate Deer Herd (Fig. 1). This area has been well described as the winter destination for thousands of migratory mule deer, *Odocoileus hemionus hemionus* (Interstate Deer Herd Committee 1951, Salwasser⁶ 1979, CDFG unpublished data). Winter range habitat types identified with landsat thematic imagery (Fox et al.⁷ 1997) were clipped and categorized using Arcview GIS 3.0a (ESRI⁸ 1992-1997). A Chi-square test (DataMost Inc.⁸) was then used to determine if the location of transects were distributed proportionally among types.

Transects were surveyed in July 1957, 1967, 1977, and 1998 using a standardized line-intercept technique (Canfield 1939, Hormay 1949). At 0.61-m intervals along each transect an observation "hit" was recorded for two categories that included ground-

⁵McDonald, G. A. 1966. Geology of the Cascade Range and Modoc Plateau. Pages 65-96 in: E. H. Bailey, editor. Geology of Northern California, California Division of Mines and Geology Bulletin 190.

⁶Salwasser, H. 1979. The ecology and management of the Devil's Garden Interstate Deer Herd and its range. PH.D. Dissertation, University of California, Berkeley, California, USA.

⁷Fox, L. III, and 6 others. 1997. A wildlife habitat map and database for the ORCA (Oregon-California) Klamath bioregion derived from landsat imagery. Technical Report Series I. The Spatial Information Systems Institute, College of Natural Resources and Sciences, Humboldt State University, Arcata, California, USA.

⁸The use of trade names does not imply endorsement by the California Department of Fish and Game.

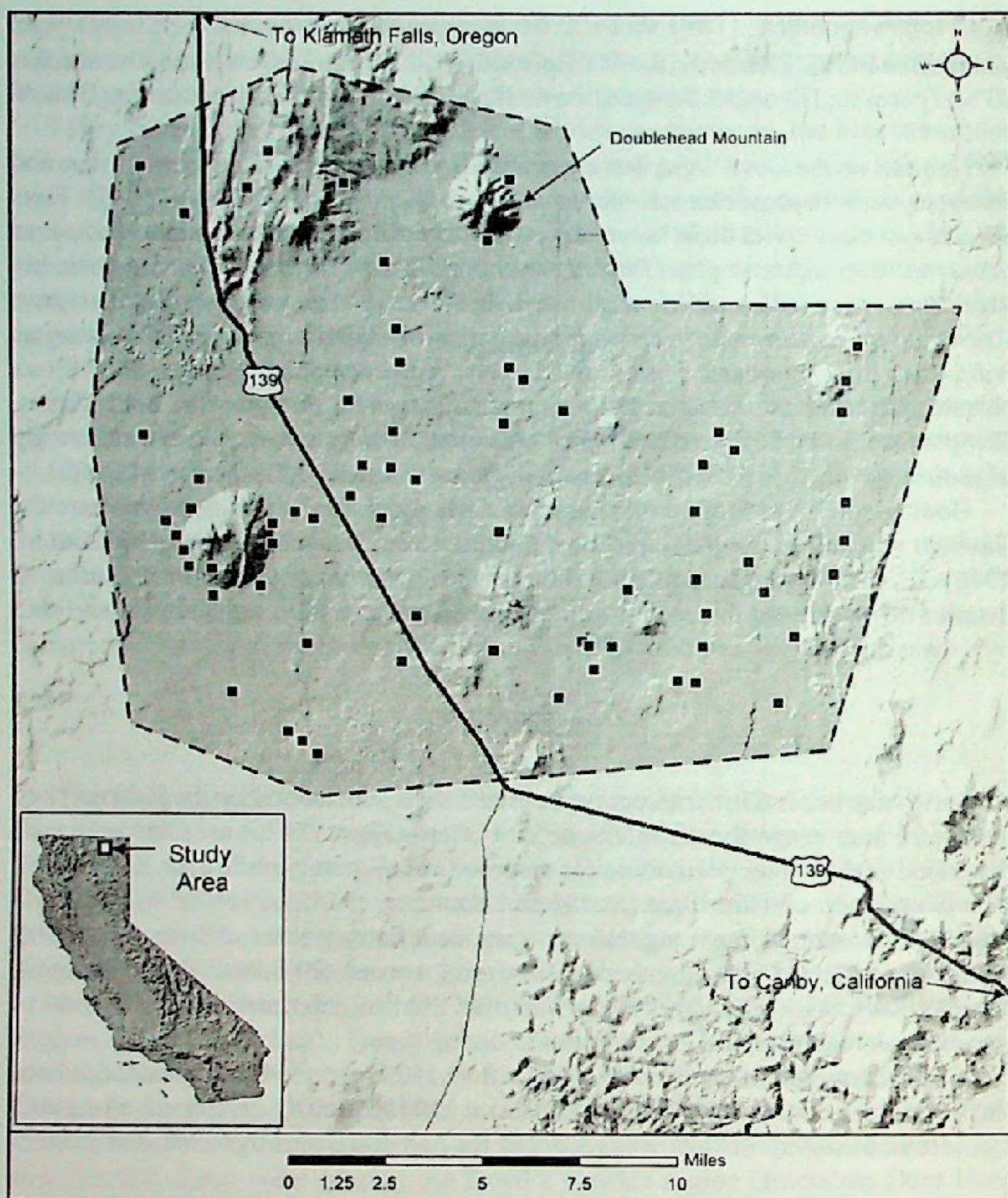


Figure 1. Study area on the Devil's Garden showing the location of vegetation monitoring transects and key area (dashed line) of deer winter range in Modoc County, California.

cover and overstory, each totaling 6900 hits per survey year. A ground-cover hit was considered the dominant vegetation within a 10.16 cm circle from the point on the transect to 1.5 m in height; overstory was the dominant vegetation > 1.5 m in height. The status (alive or dead) for ground-cover hits of all shrubs was also recorded.

Annual percent-cover for overstory and ground-cover was determined by dividing the number of hits for an individual species by the total hits of its respective category

($n = 6900$) and multiplying by 100. Annual distribution for overstory and ground-cover species was measured as a percentage by dividing the number of transects containing an individual species by the total number of transects ($n = 69$) and multiplying by 100. Species occurring in trace amounts ($< .005\%$) and herbs were excluded from analysis, with the exception for changes in the distribution of cheatgrass.

Interannual differences in the percent-cover of ground-cover and overstory species was tested with a repeated measures Anova; a Tukey multiple comparison test identified which years differed (DataMost, Inc.⁸). Changes in species distribution was tested with the Chi-square statistic (DataMost, Inc.⁸). For all statistical analysis, significance was determined at the .05 level.

RESULTS

Transects were distributed in proportion to winter range habitat types ($X^2 = 5.67$; $df = 8$; $P > 0.05$). A difference between years in the distribution of species was detected in cheatgrass ($X^2 = 40.16$; $df = 3$; $P < .0001$), dead bitterbrush ($X^2 = 31.65$; $df = 3$; $P < .001$), western juniper ground cover ($X^2 = 9.62$; $df = 3$; $P < 0.05$), and western juniper overstory ($X^2 = 20.73$; $df = 3$; $P < .001$) (Table 1).

Interannual differences in percent-cover of ground-cover included dead sagebrush ($F = 10.77$; $df = 3$; $P < .0005$), bitterbrush ($F = 9.44$; $df = 3$; $P < 0.0005$), dead bitterbrush ($F = 11.94$; $df = 3$; $P < .0005$), rabbitbrush ($F = 5.48$; $df = 3$; $P < 0.001$), mahala mat ($F = 6.96$; $df = 3$; $P < .001$) and western juniper ($F = 10.36$; $df = 3$; $P < 0.0005$) (Table 2). Bitterbrush and rabbitbrush declined from 1967 to 1998, while dead bitterbrush and dead sagebrush increased for the same period; western juniper increased for all years when compared to 1998, and mahala mat declined from 1956 to 1967 (Table 3).

For overstory, interannual differences in percent-cover occurred for western juniper ($F = 30.96$; $df = 3$; $P < .0005$) and ponderosa pine ($F = 10.93$; $df = 3$; $P < 0.0005$) (Table 2). These species increased from 1957 to 1998, with additional increases in western juniper detected from 1967 and 1977 to 1998 (Table 4).

DISCUSSION

Overall, major increases in tree overstory occurred on transects for all years as live shrubs declined and dead shrubs increased since 1967. Moreover, the increased distribution of western juniper ground-cover, overstory, and dead bitterbrush is consistent with patterns of increasing overstory canopy and reductions in shrubs. Overstory-understory relationships are well documented in many types of woodland habitats, with increases in overstory resulting in the reduction and often elimination of shrubs and herbs (Pase 1958, Moir 1966, Jameson 1967, Young et al. 1967, Tausch and Tueller 1990, Miller et al. 2000). An inverse relationship between total percent-cover of all live and dead shrubs may suggest a threshold was reached in 1967, when a 30-year decline of live shrubs and increase in dead shrubs began (Fig. 2). Declines in shrubs is further supported by the increased presence of all shrub cover in 1967 (18.5%) as compared to 1998 (13.5%). Mahala mat, a seed-banking species with a tolerance for

Table 1. Percent of transects (N = 69) containing species by year on the Devil's Garden in Modoc County, California.

<u>Live Ground Cover</u>	<u>1957</u>	<u>1967</u>	<u>1977</u>	<u>1998</u>
<i>Bromus tectorum</i> *	47.8	26.0	53.6	76.8
<i>Purshia tridentata</i>	73.9	72.4	72.4	57.9
<i>Artemesia tridentata</i>	57.9	53.6	57.9	46.3
<i>Chrysothamnus</i>	50.7	57.9	50.7	46.3
<i>Mahala mat</i>	28.9	26.0	27.5	21.7
<i>Juniperus occidentalis</i> *	43.4	40.5	24.6	49.2
<u>Dead Ground Cover</u>				
<i>Purshia tridentata</i> *	68.1	28.9	36.2	63.7
<i>Artemesia tridentata</i>	49.2	42.0	30.4	36.2
<i>Chrysothamnus</i>	24.6	11.5	17.3	13.0
<u>Overstory</u>				
<i>Juniperus occidentalis</i> *	31.8	49.2	60.8	68.1
<i>Pinus ponderosa</i>	26.0	33.3	36.2	36.2

* significant @ .05

Table 2. Annual percent-cover (hits per species / total hits) x 100 for live and dead ground-cover and tree overstory on 69 transects measured on the Devil's Garden in Modoc County, California.

<u>Live Ground-Cover</u>	<u>1957</u>	<u>1967</u>	<u>1977</u>	<u>1998</u>
<i>Artemesia tridentata</i>	4.9	5.6	5.2	4.3
<i>Purshia tridentata</i> *	5.1	7.7	6.2	4.2
<i>Chrysothamnus</i> *	2.0	2.6	1.8	1.1
<i>Ceanothus prostratus</i> *	2.7	0.9	1.4	1.6
<i>Juniperus occidentalis</i> *	1.4	1.6	1.0	3.4
<u>Dead Ground-Cover</u>				
<i>Purshia tridentata</i> *	2.7	0.5	0.9	2.5
<i>Chrysothamnus</i>	0.5	0.1	0.3	0.3
<i>Artemesia tridentata</i> *	2.6	1.0	0.7	1.4
<u>Overstory</u>				
<i>Pinus ponderosa</i> *	2.1	4.3	6.3	7.4
<i>Juniperus occidentalis</i> *	2.1	3.9	6.0	10.2

* significant @ .05

Table 3. Tukey multiple means comparison test for live and dead ground-cover species that differed among years for percent-cover on 69 transects measured on the Devil's Garden in Modoc County, California.

	<u>1957</u>		<u>1967</u>		<u>1977</u>		<u>1998</u>	
	<u>\bar{x}</u>	<u>SD</u>	<u>\bar{x}</u>	<u>SD</u>	<u>\bar{x}</u>	<u>SD</u>	<u>\bar{x}</u>	<u>SD</u>
<i>Purshia tridentata</i> (a)	5.10	5.38	7.71	8.34	6.26	6.98	4.27	5.99
Dead <i>Purshia tridentata</i> (b)	2.73	3.39	0.55	1.10	0.92	1.55	2.50	3.91
<i>Chrysothamnus</i> (c)	2.05	2.88	2.66	3.76	1.84	2.68	1.18	1.82
Dead <i>Artemesia tridentata</i> (d)	2.68	3.93	1.04	1.75	0.78	1.71	1.43	2.70
<i>Juniperus occidentalis</i> (e)	1.44	2.13	1.60	3.09	1.02	2.53	3.37	5.88
<i>Ceanothus prostratus</i> (f)	2.71	6.01	0.97	2.12	1.44	3.54	1.65	4.03

a)1967 \neq 1998
b)1957 \neq 1967 and 1977
c)1967 \neq 1998
d)1957 \neq 1967 and 1977 and 1998
e)1998 \neq 1957 and 1967 and 1977
f)1957 \neq 1967
 \bar{x} = (total hits per species / total hits per transects) x 100

Table 4. Tukey multiple means comparison test for percent-cover of tree overstory species that differed among years on 69 transects measured on the Devil's Garden in Modoc County, California.

	<u>1957</u>		<u>1967</u>		<u>1977</u>		<u>1988</u>	
	<u>\bar{x}</u>	<u>SD</u>	<u>\bar{x}</u>	<u>SD</u>	<u>\bar{x}</u>	<u>SD</u>	<u>\bar{x}</u>	<u>SD</u>
<i>Juniperus occidentalis</i> (a)	2.17	4.3	3.92	6.17	6.00	8.25	10.23	12.30
<i>Pinus ponderosa</i> (b)	2.20	5.15	4.27	7.80	6.34	10.76	7.59	13.09

a)1957 \neq 1977 and 1998; 1967 \neq 1977 and 1998.
b)1957 \neq 1998.
 \bar{x} = (total hits per species / total hits per transects) x 100

shade, followed a similar pattern for percent-cover as dead shrubs that may be related to variations in the detection of this prostrate growing species. Given the comprehensive sampling of this winter range and the equal distribution of transects relative to habitat types, it is reasonable to assume that patterns of change in ground-cover and tree overstory are representative of current trends in vegetation on the Devil's Garden.

The greatest change occurred in western juniper, as it was the only species to increase on transects in ground-cover, overstory, and distribution. The large increase in western juniper overstory since 1957 may suggest that a decline in its ground-cover in 1977 was related to the maturation and conversion of this conifer from ground-cover to overstory. Western juniper is capable of dominating all sites within the ecological

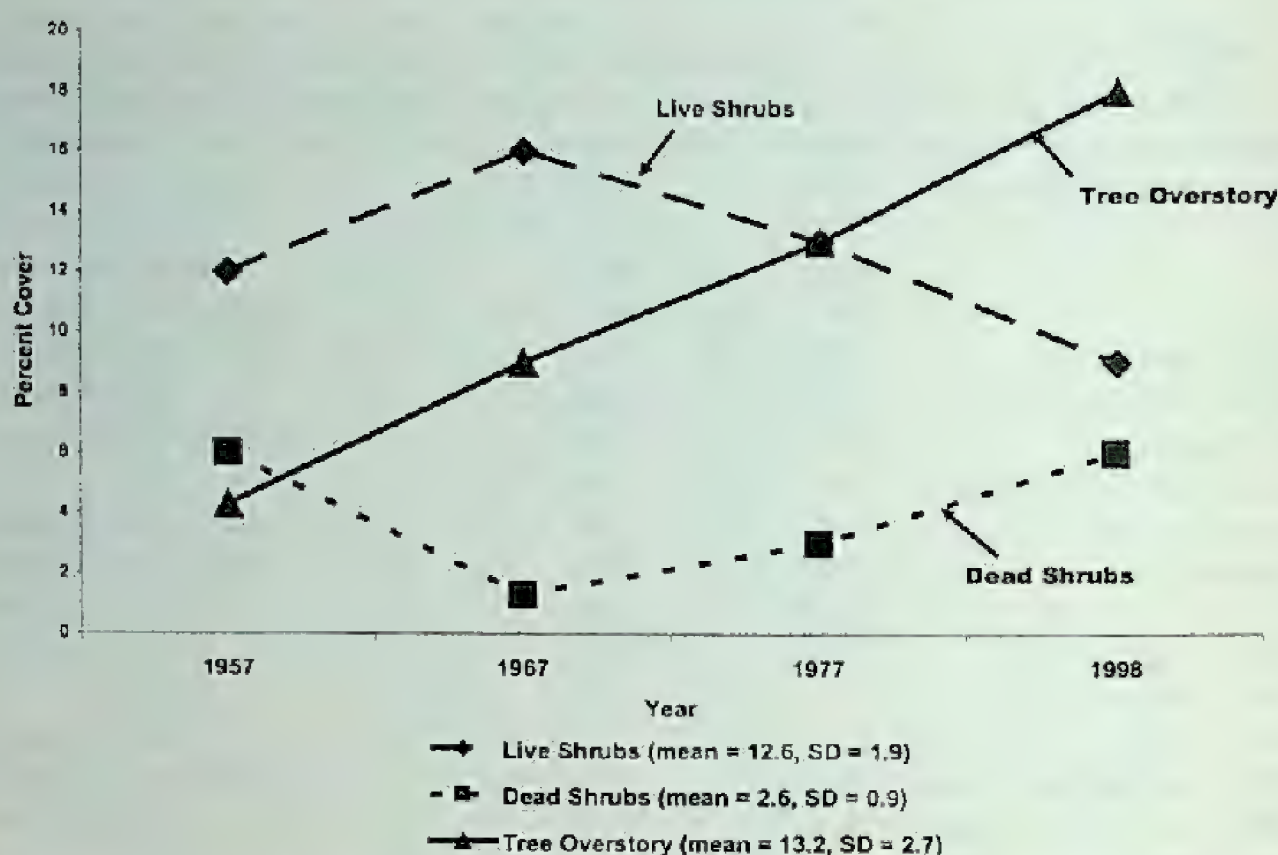


Figure 2. Annual percent-cover (hits per species / total hits) $\times 100$ for live and dead ground-cover shrubs (big sagebrush, bitterbrush, rabbitbrush) and tree overstory (ponderosa pine and western juniper) measured on 69 transects on the Devil's Garden in Modoc County, California.

belt to which it is confined (Evans and Young 1985, Goodrich and Reid⁹ 1999). The encroachment and aging of a western juniper woodland can reduce soil moistures (Breshears et al. 1997), and severely limit the hydrologic regime of watersheds (McCarthy and Dobrowolski¹⁰ 1999). Increased soil erosion and sediment delivery on a western juniper dominated landscape is typical of older sites where low nutrient content, excessive water repellency, and levels of gypsum or other chemicals can inhibit plant growth (Goodrich and Reid⁹ 1999). The elimination of shrubs and herbs important to wildlife is well documented with the establishment and progression of a western juniper stand (Arnold 1964, Blackburn and Tueller 1970, Schott and Pieper 1985, Tausch et al. 1981, Tausch and Tueller 1990, Miller et al. 2000). The dramatic increase of this fire-intolerant species on the Devil's Garden may be having profound impacts on this region's ecological potential.

A significant increase in the distribution of cheatgrass may be similar to other Great

⁹Goodrich S. and C. Reid. 1999. Pages 317-321 in: S. B. Monsen and R. Stevens, compilers. Proceedings: ecology and management of pinyon-juniper communities within the Interior West: 1997 September 15-18: Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

¹⁰F. J. McCarthy and J. P. Dobrowolski. 1999. Pages 194-199 in: S. B. Monsen and R. Stevens, compilers: Proceedings: ecology and management of pinyon-juniper communities within the Interior West: 1997 September 15-18: Provo, UT. Proc. RMRS-P-9. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Basin rangelands where the introduction and encroachment of this exotic annual has occurred (Young et al. 1972). Its reduction on transects in 1967 may be related to annual differences in amounts and timing of precipitation (Sneva and Hyder 1962). The invasive characteristics of cheatgrass can convert native vegetation to fire-maintained annual grass rangelands (Young et al. 1972, Loft¹¹ 1990); impacting wildlife habitat values (McAdoo and Klebenow¹² 1978, Mack 1981) and populations (Gano and Rickard 1982, Groves and Steenhof 1988, Loft¹¹ 1990).

MANAGEMENT IMPLICATIONS

Investigation of trends in Great Basin vegetation have often relied on indirect or short-term techniques to describe long-term changes in habitats. Some researchers have used historical photos, anecdotal accounts, or present-day sampling methods to describe trends in the seral condition of Great Basin habitats (Shinn 1980, Tausch et al. 1981, Arno and Gruell 1983, Gruell¹ 1986). Peek et al. (2000) used aerial and satellite imagery and a geographical information system to document increases in forest canopy closure and significant reductions in understory biomass from 1953 to 1988. Miller and Rose (1999) analyzed ring and core samples from western juniper and ponderosa pine to study the post-settlement expansion of western juniper. This study provides evidence of a successional advancement in habitats using direct measurements of vegetation to quantify temporal changes in ground-cover and tree overstory.

There may be compelling evidence that patterns in shrub communities are associated with trends in shrub-dependent wildlife populations. The mid-20th century eruption of mule deer has largely been attributed to vast increases in the abundance of shrubs (Leopold et al. 1947, Caughley 1969, Gruell¹ 1986). On adjacent summer range in Oregon, Peek et al. (2000) concluded that population declines of the Interstate Deer Herd were associated with a shift towards closed-canopy forests and a 34.5% decrease in understory biomass production. On the Devil's Garden, reductions in mule deer (Salwasser⁶ 1979) and live shrubs beginning in late 1960's may suggest a relationship between shrub availability and the severe long-term decline of these native herbivores.

Sage grouse, *Centrocercus urophasianus*, another shrub-obligate species native to the Great Basin (Braun et al. 1977), reached peak population levels in the 1930's, but dramatic declines on the Devil's Garden since the 1960's (CDFG unpublished data) is indicative of a range-wide trend prompting its potential listing as threatened or endangered. Adjoining the Great Basin in the shrub-steppe habitats of the interior Columbia River basin, shrub nesting bird species have declined since 1968, whereas

¹¹Loft, E. R. and J. W. Menke. 1990. Evaluation of fire effects on mule deer habitat in Lassen County. Final report. California Department of Fish and Game contract FG1c-2090. University of California, Davis, Department of Agronomy and Range Science, 30p.

¹²McAdoo, J. K. and D. A. Klebenow. 1978. Native faunal relationships in the sagebrush ecosystem. Pages 50-61 in: The sagebrush ecosystem. A symposium. Logan, Utah. Utah State University, College of Natural Resources.



Photo-point comparison (facing pages) of a vegetation monitoring transect showing changes in juniper/shrub-steppe habitat from 1948 (top photos) to 1998 (bottom photos) on the Devil's Garden, Modoc County, California (arrow indicates stake used to identify transect).



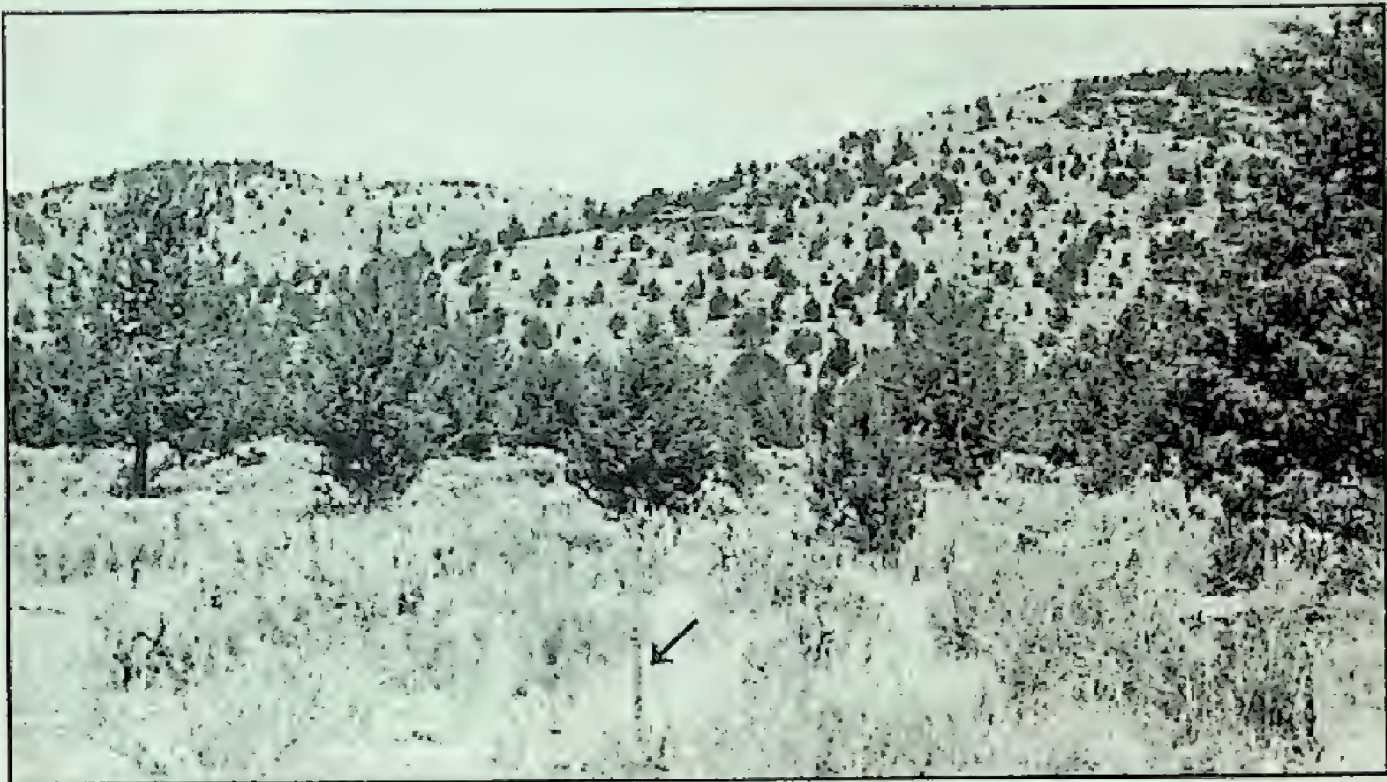




Photo-point comparison (facing pages) of a vegetation monitoring transect showing changes in eastside pine habitats from 1948 (top photos) to 1998 (bottom photos) on the Devil's Garden, Modoc County, California (arrow indicates stake used to identify transect)





the canopy nesters have increased (Saab and Rich¹³ 1997). A recent study on the Devil's Garden suggests that as western juniper woodlands become increasingly dense, and the shrub layer declines, avian abundance and diversity is ultimately reduced (Miller¹⁴ 2001). The succession of habitats may be an important consideration when developing strategies for the management and conservation of wildlife requiring a diversity of seral conditions.

Lastly, across many western ranges, shrub stands continue to grow senescent (Gruell¹ 1986, Urness¹⁵ 1990, Peek et al. 2000). Many Great Basin shrub stands were established > 100 years ago and are declining due to natural maturation (Clements and Young 1997), or inability to rejuvenate after fire (Gruell¹ 1986). On the Devil's Garden, shrubs increased following the 19th century introduction of livestock grazing (Salwasser⁶ 1979), and may have reached highest levels of abundance and productivity by the 1960's. Maximum longevity and foliage yields for bitterbrush and sagebrush have been reported at <100 years (Hormay¹⁶ 1943, Lommasson 1948, Ferguson 1964, Roughton 1972, Daubenmire 1975, McConnell and Smith 1977, Young and Palmquist 1992), and the extirpation of these species from the Devil's Garden (D. Thayer, personal observation) and many other Great Basin rangelands have resulted from fire (Blaisdell and Mueggler 1956, Nord 1965, Ralphs and Busby 1979, Wambolt and Payne 1986, Loft¹¹ 1990). Shrub stands on the Devil's Garden may become increasingly evanescent as the cumulative effects of natural maturation, cheatgrass invasion, and competition from tree overstory dominance further reduce the vigor and resilience of these habitats.

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¹³Saab, V. A. and T. D. Rich. 1997. Large-scale conservation assessment for neo-tropical migratory land birds in the interior Columbia River Basin. United States Department of Agriculture and United States Department of Interior, General Technical Report PNW-GTR-399.

¹⁴Miller, R. F. 2001. Managing western juniper for wildlife. Woodland Fish and Wildlife. Publication number misc0286.

¹⁵Urness, P. J. 1990. Livestock as manipulators of mule deer winter range habitats in northern Utah. United States Forest Service General Technical Report RM-194, pp. 25-40.

¹⁶Hormay, A. L. 1943. Bitterbrush in California. Research Note 34. United States Department of Agriculture, Forest Service, Berkeley, California.

from the hunter-conservationists of California, the Deer Herd Management Plan Implementation Program, and the Wildlife Programs branch of the Northern California-North Coast Region.

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THE BIRD COMMUNITY OF AN OAK WOODLAND STREAM

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The oak woodlands of California provide critical habitat for a number of avian species. Within the oak landscape, riparian zones, a highly restricted habitat element sensitive to alterations, provide unique subtle qualities that are exploited by both resident and migratory species. We provide information from 10 years of bird census data collected and categorized from Parson's Creek, an ephemeral tributary of the Russian River in Mendocino County. The data has been collected bi-annually at 12 permanent observation stations, distributed along a 1-mile section of the creek. We provide a synopsis of data collected between spring 1993 and spring 2002. A total of 1,912 observations has been collected identifying 81 species, representing 33 taxonomic families and 9 orders. A list of species, their migratory strategies and season of occurrence is provided.

INTRODUCTION

Riparian areas have long been recognized as being important to both resident and migratory bird species from a variety of xeric habitats throughout the western United States (Carothers and Johnson 1975; Gaines 1977; Wauer 1977; Bull 1978; Szaro 1980). Giusti et al. (1996) predicted potential avian assemblages in oak woodland riparian habitats using existing models (Mayer and Laudenslayer 1988) as a means to help demonstrate the importance of riparian considerations in management actions. However, few accounts exist for locally important oak woodland riparian habitats in north coastal California that demonstrate or validate avian assemblages over time.

During the past few years, concerns have been expressed about the influence of land-use practices on both oak woodlands and their riparian habitats along the north coast of California (Merenlender 1998). Recently, scientific information detailing the impacts of land-use changes on the oak woodland biota has become available illustrating the ecological consequences of habitat fragmentation (Merenlender et al. 1998), and the importance of maintaining functional habitat elements within an altered landscape (Hilty 2001).

As intensive agriculture and urban sprawl continue to alter lands that have historically been used for low-intensity agriculture (livestock production) the ecological

integrity of oak woodland communities has been called into question (Merenlender et al. 1998; Walter 1998; Mummert et al. 2002; Stralberg and Williams 2002). As encroachment further exacerbates degraded conditions within oak woodland riparian zones, field studies will be useful in guiding land-use activities and projects that may impact stream zones. Unfortunately, most conservation projects are done in less than 5 years (Dobson and Lyles 1989) thereby limiting their ability to interpret long-term trends. In the face of a rapidly changing landscape (Heise and Merenlender 2002) the need for long-term, scientifically valid field data should be obvious to address issues between competing interests groups involved in land-use discussions.

STUDY SITE

The study site is a 1-mile section of Parson's Creek, an ephemeral tributary of the Russian River on the University of California Hopland Research and Extension Center property in Mendocino County. The stream typically has surface water flowing from late autumn (November) through early summer (June). However, at any time throughout the summer water can be found in deep, shaded pools. We used the California Wildlife Habitat Relationships model (CWHIR) to classify streamside vegetation. The dominant vegetation types within the study site are blue oak woodland, and valley oak riparian habitats (Mayer and Laudenslayer 1988). The dominant woody plant species found within the study site include: blue oak (*Quercus douglassi*), valley oak (*Q. lobata*), Oregon white oak (*Q. garryana*), interior live oak (*Q. wislizenii*), white alder (*Alnus rhombifolia*), willow (*Salix* sp) and blackberry (*Rubus procerus* var. *Himalaya*).

Since the site traverses a 1-mile path, it is surrounded by a variety of other oak dominated vegetation types. Additionally, portions of the creek (and the subsequent data collection points) are adjacent to both vineyard and livestock agricultural operations. In some reaches of the stream, livestock and deer have been excluded from the creek affecting the distribution of vegetation cover within the study site.

METHODOLOGY

Bird census data was collected using the point-count method first described in the French literature by Ferry and Frochet (1970). The methodology was later defined in the English literature by (Jorgensen 1974) who credited this method with providing: 1) a standardized approach to data collection, 2) permanently established data collection points, 3) a protocol that insures surveyor stability, and 4) time efficiency.

Bird count data was collected within the stream corridor including: 1) the stream channel, 2) the flood channel, and 3) the vegetation found growing in the channel and immediately adjacent to the water (Tubbs 1980). Birds were counted if they occurred on the ground, in the water, in any snags, trees or other vegetation found within the defined stream channel.

The stream is currently divided into 12 sections of approximately equal lengths, from an earlier, unrelated project, with fences that exclude sheep and deer. Since the

exclusionary fencing is not in a randomize fashion no attempt was made to compare ungulate exclusion or vegetation recovery with bird presence or absence. We felt this approach best mimics local demographics where changing ownership patterns and management strategies are commonplace (Hilty and Merenlender 2003). For our purposes, a permanent sampling point was arbitrarily selected and located within each of the existing sections. The sampling points are located in an area that provides the observer and data recorder the best vantage point from which to identify birds within the delineated section. Species were identified by sight and/or call.

The 12 stations have been consistently monitored bi-annually during spring and fall migratory periods between 1993 and 2002. Each station is sampled for 20 minutes per census period, twice in the spring and once in the fall. All sampling is done in the mornings after sunrise. The time spent per day surveying birds equals 3 hours. Thus, the total number of hours per year spent surveying birds is 9 hours (6 hours in spring, 3 hours in fall). A total of 90 hours of observations has been amassed between 1993 and 2002.

The two spring census periods coincided with the arrival of migratory songbirds between mid-May and early June. The fall census is conducted in mid-October. Only those birds exhibiting active behaviors (perching, foraging, singing, nesting, etc) within the riparian corridor are tallied. Birds flying high above the stream zone are not counted.

RESULTS

A total of 1,912 bird observations has been tallied identifying 81 species, representing 33 taxonomic families and nine orders (Table 1). The number of individual bird sightings observed per year ranged from a high of 254 (1993) to a low of 173 (1999). The average number of individuals recorded per year during the 10-year assessment is 198. (Fig. 1).

Seasonal species assemblages were identified with some species demonstrating a consistent pattern of seasonal occurrence. Twenty-nine species (ranging from 27 to 33) were detected during the fall counts compared to 43 species (ranging from 38 to 48) in the spring (Fig. 2). The shift in seasonal avian assemblage is considered reflective of the migratory strategies of many of the birds observed. Of the 81 species detected, 24 were observed only during the spring period, 13 only during the fall counts, and 44 species were detected in both seasons. Of these, 52 are considered resident species, 8 are considered short distant migrants and 21 are classified as neo-tropical migratory songbirds [(Sibley 2000) (Table 1)].

CONCLUSIONS

The list of birds making use of the stream corridor between 1993 and 2002 is consistent with predicted diversity indices that have been projected for oak woodland stream corridors. Some species have demonstrated a strong seasonal preference for oak woodland riparian areas while others can be found in the riparian corridor throughout

Table 1. List of birds from Parson's Creek, Hopland Research and Extension Center, Hopland, California. 1993-2002. Classification of migratory status: R = resident. SD = Short Distance Migrant, NT = Neotropical Migrant. Season of Occurrence: S = spring. F = fall, B = both fall and spring.

Common Name	Scientific Name	Migratory Strategy	Season of Occurrence
Order: Ciconiiformes			
Family: Ardeidae			
Great Blue Heron	<i>Ardea herodias</i>	R	F
Great Egret	<i>Ardea alba</i>	R	F
Order: Anseriformes			
Family: Anatidae			
Mallard	<i>Anas platyrhynchos</i>	R	S
Wood Duck	<i>Aix sponsa</i>	R	S
Order: Falconiiformes			
Family: Cathartidae			
Turkey Vulture	<i>Cathartes aura</i>	R	S
Family: Accipitridae			
Sharp-shinned Hawk	<i>Accipiter striatus</i>	R	S
Red-shouldered Hawk	<i>Buteo lineatus</i>	R	S
Red-tailed Hawk	<i>B. jamaicensis</i>	R	S
Family: Faclonidae			
American Kestrel	<i>Falco sparverius</i>	R	B
Order: Galliformes			
Family: Odontophoridae			
California Quail	<i>Callipepla californica</i>	R	B
Family: Phasiandidae			
Wild Turkey	<i>Meleagris gallopavo</i>	R	B
Order: Charadriiformes			
Family: Charadriidae			
Killdeer	<i>Charadrius vociferus</i>	R	S
Order: Columbiformes			
Family: Columbidae			
Mourning Dove	<i>Zenaida macroura</i>	R	B
Order: Apodiformes			
Family: Trochilidae			
Anna's Hummingbird	<i>Calypte anna</i>	R	B

Order: Piciformes**Family: Picidae**

Acorn Woodpecker	<i>Melanerpes formicivorus</i>	R	B
Downy Woodpecker	<i>Picoides pubescens</i>	R	B
Nuttall's Woodpecker	<i>Picoides nuttallii</i>	R	B
Northern Flicker	<i>Colaptes auratus</i>	R	B

Order: Passeriformes**Family: Tyrannidae**

Western Wood-Pewee	<i>Contopus sordidulus</i>	NT	S
Pacific-slope Flycatcher	<i>Empidonax difficilis</i>	NT	B
Willow Flycatcher	<i>E. traillii</i>	NT	F
Black Phoebe	<i>Sayornis nigricans</i>	R	B
Say's Phoebe	<i>S. saya</i>	SD	F
Ash-throated Flycatcher	<i>Myiarchus cinerascens</i>	NT	S
Western Kingbird	<i>Tyrannus verticalis</i>	NT	S

Family: Vireonidae

Warbling Vireo	<i>Vireo gilvus</i>	NT	B
Hutton's Vireo	<i>V. huttoni</i>	R	B
Cassin's Vireo	<i>V. cassinii</i>	NT	B

Family: Corvidae

Stellar's Jay	<i>Cyanocitta stelleri</i>	R	B
Western Scrub Jay	<i>Aphelocoma californica</i>	R	B
American Crow	<i>Corvus brachyrhynchos</i>	R	B

Family: Hirundinidae

No. Rough-winged Swallow	<i>Stelgidopteryx serripennis</i>	NT	S
Violet-green Swallow	<i>Tachycineta thalassina</i>	NT	S
Tree Swallow	<i>T. bicolor</i>	NT	S

Family: Paridae

Oak Titmouse	<i>Baeolophus inornatus</i>	R	B
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Family: Aegithalidae

Bushtit	<i>Psaltiriparus minimus</i>	R	B
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Family: Sittidae

White-breasted Nuthatch	<i>Sitta carolinensis</i>	R	B
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Family: Certhiidae

Brown Creeper	<i>Certhia americana</i>	R	B
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Family: Troglodytidae

Bewick's Wren	<i>Thryomanes bewickii</i>	R	F
House Wren	<i>Troglodytes aedon</i>	SD	B

Family: Timaliidae			
Wrentit	<i>Chamaea fasciata</i>	R	F
Family: Regulidae			
Ruby-crowned Kinglet			
Regulus calendula	SD	F	
Family: Sylviidae			
Blue-gray Gnatcatcher	<i>Polioptila caerulea</i>	NT	S
Family: Turdidae			
Western Bluebird	<i>Sialia mexicana</i>	R	B
American Robin	<i>Turdus migratorius</i>	R	B
Hermit Thrush	<i>Catharus guttatus</i>	R	F
Family: Mimidae			
Northern Mockingbird	<i>Mimus polyglottos</i>	R	F
Family: Sturnidae			
European Starling	<i>Sturnus vulgaris</i>	R	B
Family: Bombycillidae			
Cedar Waxwing	<i>Bombycilla cedrorum</i>	SD	F
Family: Parulidae			
Orange-crowned Warbler	<i>Vermivora celata</i>	R	B
Nashville Warbler	<i>V. ruficapilla</i>	NT	S
Yellow Warbler	<i>Dendroica petechia</i>	NT	B
Yellow-rumped Warbler	<i>D. coronata</i>	R	B
Black-throated Gray Warbler	<i>D. nigrescens</i>	NT	S
Townsend's Warbler	<i>D. townsendi</i>	NT	S
Common Yellowthroat	<i>Geothlypis trichas</i>	R	F
Wilson's Warbler	<i>Wilsonia pusilla</i>	NT	B
Family: Thraupidae			
Western Tanager	<i>Piranga ludoviciana</i>	NT	B
Family: Cardinalidae			
Black-headed Grosbeak	<i>Pheucticus melanocephalus</i>	NT	S
Lazuli Bunting	<i>Passerina amoena</i>	NT	S
Family: Emberizidae			
Spotted Towhee	<i>Pipilo maculatus</i>	R	B
California Towhee	<i>P. crissalis</i>	R	B
Rufous-crowned Sparrow	<i>Aimophila ruficeps</i>	R	B
Chipping Sparrow	<i>Spizella passerina</i>	NT	B
Lark Sparrow	<i>Chondestes grammacus</i>	R	B
Golden-crowned Sparrow	<i>Zonotrichia atricapilla</i>	SD	B

White-crowned Sparrow	<i>Z. leucophrys</i>	SD	B
Fox Sparrow	<i>Passerella iliaca</i>	R	F
Song Sparrow	<i>Melospiza melodia</i>	R	F
Lincoln's Sparrow	<i>M. lincolnii</i>	SD	B
Dark-eyed Junco	<i>Junco hyemalis</i>	R	B
Family: Icteridae			
Western Meadowlark	<i>Sturnella neglecta</i>	R	S
Brown-headed Cowbird	<i>Molothrus ater</i>	R	S
Red-winged Blackbird	<i>Agelaius phoeniceus</i>	R	B
Brewer's Blackbird	<i>Euphagus cyanocephalus</i>	R	B
Bullock's Oriole	<i>Icterus bullockii</i>	NT	S
Family: Fringillidae			
Purple Finch	<i>Carpodacus purpureus</i>	R	B
House Finch	<i>C. mexicanus</i>	R	B
Lawrence's Goldfinch	<i>C. lawrencei</i>	SD	S
Lesser Goldfinch	<i>C. psaltria</i>	R	B
American Goldfinch	<i>C. tristis</i>	R	S

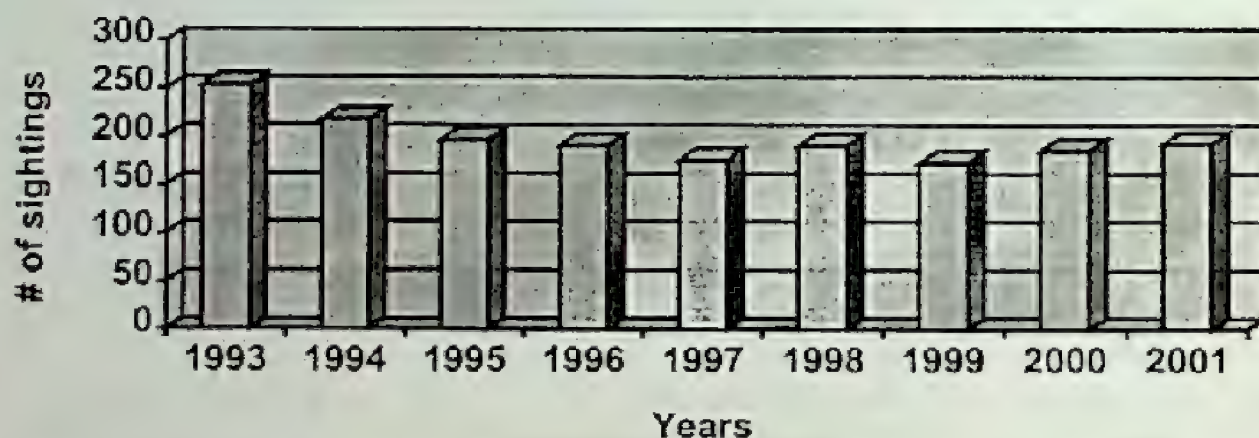


Figure 1. Number of sightings, Parson's Creek, 1993-2001. (n=1779, \bar{x} =198/yr., Rng=254-173)

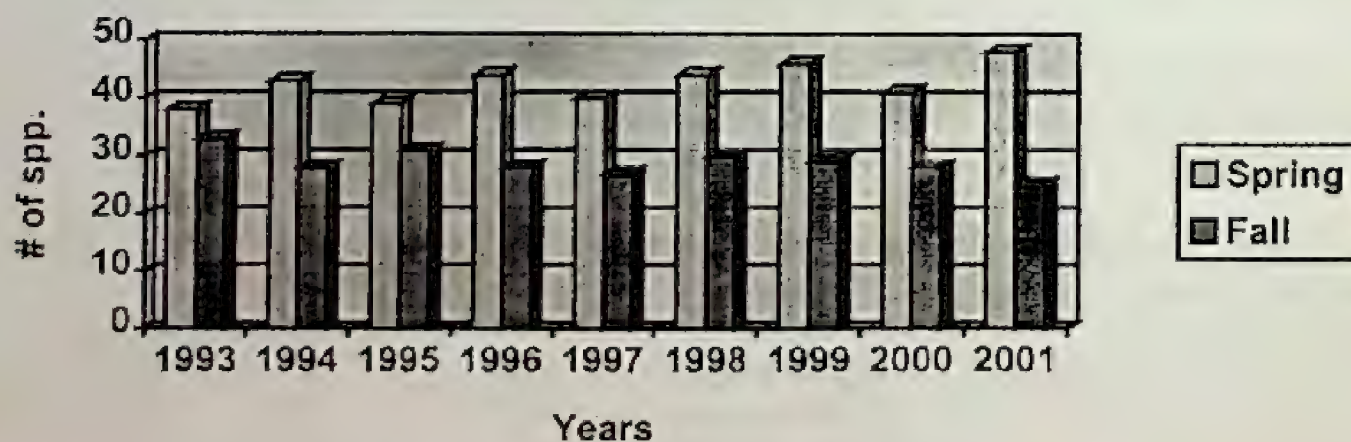


Figure 2. Number of species observed, Paron's Creek, 1993-2001.

the year. Though this study and its data will be subjected to further analysis and scrutiny, this type of field-validated data will be useful in demonstrating the relationship between riparian habitats and songbird utilization to land owners and other decision makers on the north coast.

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HABITAT ASSOCIATIONS OF STEELHEAD TROUT NEAR THE SOUTHERN EXTENT OF THEIR RANGE

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Habitat associations of wild steelhead trout, *Oncorhynchus mykiss*, in southern and south-central California are poorly described in the scientific literature and are largely unknown. Descriptive surveys and a manipulative experiment were performed to assess use of depth, velocity, run and pool by wild age-0, age-1 and age-2 steelhead in a small south-central California stream. Juvenile steelhead showed size-specific use of depth and velocity, though water depth was the better predictor of size-specific microhabitat use. Age-0 steelhead used shallow areas of the stream, whereas age-1 and age-2 steelhead used relatively deep areas. Juvenile steelhead used pools and runs, but age-0 steelhead showed greater occupancy of runs than was expected when abundance was adjusted according to the size of the sampled unit. Spatial variation in the density of juvenile steelhead was related to the water depth and size of the sampled units and the amount of riparian canopy cover. Because the habitats used by age-0 steelhead may not be suitable for age-1 and older steelhead, the results suggest that multiyear freshwater residence of steelhead requires habitat that is appropriate for all age classes.

INTRODUCTION

Abundance of wild steelhead trout, *Oncorhynchus mykiss*, declined over the last several decades as a result of habitat loss, degradation and fragmentation (Nehlsen et al. 1991, Swift et al. 1993). Once distributed as far south as northern Baja California, the southern extent of steelhead in streams now appears to be southern California (32-33° N). The decline in steelhead, especially at the southern extent of their range, prompted listing of the species (National Marine Fisheries Service 1997¹) under the federal Endangered Species Act of 1973. For steelhead, in particular near the southern extent of their range, effective protection, conservation, and recovery of this species are challenged by a lack of region-specific information concerning abundance, distribution, life history, and habitat associations.

Environmental conditions and processes in southern regions differ from the conditions and processes throughout northerly areas of the species' range that extends to the southern drainages of the Alaska Peninsula (Behnke 1992). Unregulated

¹National Marine Fisheries Service. 1997. Endangered and threatened species: listing of several evolutionary significant units (ESUs) of West Coast steelhead. Federal Register 62 (159):43937-43953.

streams in south-central California can show extreme variations in discharge and temperature. Winters are mild and short in duration. Most precipitation is in the form of rainfall and typically occurs during late November through March; the remainder of the year can be hot and dry. If surface water is present during late summer or early fall, it can be shallow and intermittent, and pools can be the only suitable retreat for juvenile steelhead. In addition, *O. mykiss* is the only anadromous salmonid species in this region; interactions with other species of *Oncorhynchus* common to northern streams do not occur. The microhabitat (depth and velocity) and mesohabitat (run and pool) use by juvenile steelhead in such environmental conditions is poorly understood.

This study assessed use of microhabitat and mesohabitat by three age groups of wild juvenile steelhead in a small coastal stream in south-central California during May-September 2000 and 2001. The objectives were to describe age and size-specific use of various mesohabitat types (run and pool) and microhabitat characteristics (depth and velocity), and to assess relationships between density of juvenile steelhead and various habitat properties (depth, mesohabitat area, riparian canopy and density of older conspecifics).

STUDY AREA

Santa Rosa Creek, San Luis Obispo County, California is a small coastal drainage (49.6 km²) that directly enters the Pacific Ocean and extends to an elevation of 700 m (Fig. 1). Oak, *Quercus* spp., woodland is present in discrete patches throughout the watershed, though grassland dominates the slopes. The only urban area (downtown Cambria) lies near the creek mouth. The study site was 800-m long and located 18 km from the ocean at an elevation of 200 m. The Creek is rocky and shallow; gravel and cobble are the principal substrata. Width of the wetted channel averaged 2.7 m. Mesohabitat is primarily run and pool; few riffles were observed. Average water depth (± 1 SD) was 24.7 (10.7) cm in pools and 13.7 (3.1) cm in runs; maximum depth was 90 cm. White alder, *Alnus rhombifolia*, California sycamore, *Platanus racemosa*, and California laurel, *Umbellularia californica*, are common within the study site, as are instream rootwads and downed trees. During May-September 2000 and 2001, discharge ranged from 0.01-0.03 m³/s. The range of water clarity and temperature was 1.8-3.4 m and 15-20.5°C.

METHODS

Use of Water Depth and Velocity

I used direct underwater observation to locate steelhead in randomly selected mesohabitat units during surveys performed in spring (May-early June; $n = 9$ for pools and $n = 10$ for runs) 2000 and summer (late July; $n = 11$ pools) 2001. The study objectives and scope necessitated the sampling schedule. Spring surveys focused on age-0 steelhead, whereas summer surveys included all age groups, but were confined to pools. Water column depth (cm), height (cm) of the fish above the channel bed (focal



Figure 1. Location of the study reach on Santa Rosa Creek, San Luis Obispo County.

point), and water velocity (cm/s) at the focal point were measured for a total of 225 individual age-0 steelhead in spring and a total of 121 age-0 and older juvenile steelhead in summer. The protocol for locating fish and estimating total length and measuring depth and velocity followed methods described in Spina (2000). Immediately after snorkeling an individual unit, availability of water depth and velocity (at 3 cm above the channel bed; the modal focal-point height) was measured at a minimum of 15 locations in each unit; locations were selected randomly following the x-y coordinate method (Baltz et al. 1991, Whalen and Parrish 1999).

Unit-specific means of use and availability of water depth and velocity in spring were calculated for each mesohabitat unit, and then submitted to a paired-sample *t* test to test the null hypothesis of no difference (type I error rate = 0.05) in mean depth and velocity between locations used by age-0 steelhead and those available. This test was performed separately for run and pool. I estimated the effect of mesohabitat type on

microhabitat selection by performing a two-sample *t* test on the difference in the microhabitats used by age-0 steelhead and the available microhabitats between run and pool, based on unit-specific values. The paired-sample *t* test was used to test the null hypothesis of no difference between summer use of depth and velocity for each age group and microhabitat availability, based on unit-specific means (total degrees of freedom for age 2 steelhead = 9 because age-2 steelhead were observed in all but one pool). Simple linear regression analysis was used to test the null hypothesis of no relationship between fish length and water depth and velocity at locations used in summer. The appropriateness of the regressions was assessed using standard methods (Wilkinson and Coward 1998). Because the error variance for the regression of velocity and length was heterogeneous even after transformation, weighted least squares was ultimately used to model this relationship.

Use of Run and Pool

The removal-depletion method (Riley and Fausch 1992) and one backpack electroshocker were used during early September 2000 to estimate abundance of steelhead in a total of 10 mesohabitat units ($n = 5$ for both pool and run). Block nets were installed to enclose each unit, and at least three passes in an upstream direction were performed at each unit. Steelhead were measured (mm, fork length) and weighed (g), scales were removed from 42 individuals, and all fish were released. The depth (cm), length (m), width, and amount of riparian canopy cover (%) were measured at a minimum of 12 randomly selected locations within each unit. Riparian canopy cover was measured using a hand-held mirror (with equal area quadrants) and level. Unit-specific means of depth, width, and riparian canopy cover were calculated from the individual values. Length-frequency distributions were prepared to identify age groups, and the age breaks were validated with scale readings using standard methods (Jerald 1983). Unit-specific estimates of fish abundance (number of fish) were calculated using MicroFish 3.0 (Van Deventer and Platts 1989).

A chi-square goodness-of-fit test (corrected for continuity, Zar 1996) was used to test the hypothesis that the proportion of an age group in run and pool was not different than an expected proportion based on mesohabitat area. The estimates of fish abundance in each unit were pooled to obtain the observed values for each age group and each mesohabitat type. Expected values for each age group in run and pool were calculated by multiplying the total area of a particular mesohabitat type with the number of fish observed (i.e., the electrofishing estimates) in the subject mesohabitat type. I used multiple linear regression analysis to assess relationships between the estimated density (fish/m²) of age-0 and age-1 and older steelhead (i.e., density estimate for combined catch of both age-1 and age-2 steelhead) in each unit and the area (m²), mean depth, and amount of riparian canopy cover (%) for each unit. The density of age-2 steelhead was included in the set of predictor variables when density of age-0 steelhead was used as the response variable. Regression diagnostics were performed for multiple regression using standard methods (Wilkinson and Coward 1998).

Colonization Experiment

I created 12-temporary pools (with two treatments: shallow and deep, $n = 6$ for each treatment) in the creek to test the null hypothesis that the abundance and size of steelhead colonizing the pools did not differ between treatments. Pools were created in glides, and a shallow treatment was always paired with a deep treatment in the same general location. Treatments were created in an upstream-downstream direction and were alternated over space. The maximum depth of shallow and deep pools ranged from 12-16 cm and 30-40 cm; average depth was 9.8 and 23.6 cm. The pools lacked physical cover and were similar in length (2.7 m) and width (1.5 m). The pools were created in late June 2000 and then revisited 5 weeks later to collect all the fish (using a backpack electroshocker and dip net) that colonized the pools. All fish were measured (mm, fork length), weighed (g), and then released. I used a paired-sample t test (treatment types paired by location) to test for differences in means of density (fish/m²), maximum length, weight (g), and biomass (g/m²) of steelhead between the treatments. Pool-specific means were used in the analyses. Power was calculated following Zar (1996). The results of the colonization experiment provided additional information for interpreting the habitat use patterns documented in the descriptive surveys.

RESULTS

Use of Water Depth and Velocity

In spring, age-0 steelhead used locations characterized by shallow, low-velocity water (Fig. 2). For both mesohabitat types combined, the average (± 1 SD) water depth and velocity used by age-0 steelhead was 17.5 (8.4) cm and 6.6 (7.1) cm/s. In pools, depths at locations used were shallower than if they had been randomly selected from the available depths ($t = 2.6$; $df = 8$; $P = 0.03$), but water velocities at locations used were not different than the distribution of available velocities ($t = 1.0$; $df = 8$; $P = 0.3$). In runs, age-0 steelhead used locations that were deeper than what would have been used if depth had been randomly selected ($t = 7.4$; $df = 9$; $P < 0.0005$). The distribution of water velocities at locations used was not different from the distribution of velocities that was available ($t = 2.0$; $df = 9$; $P = 0.08$). The depth (separate variance $t = 4.4$; $df = 9.2$; $P = 0.002$) and velocity (pooled variance $t = 2.1$; $df = 17$; $P = 0.04$) at locations used by age-0 steelhead depended on mesohabitat type.

During summer in pools, age-0 steelhead used depths and velocities that were on average 25.7 (13.1) cm and 1.4 (2.4) cm/s. Generally, age-1 and age-2 steelhead used locations characterized by relatively deep water with low velocity (Fig. 2). The depths and velocities at locations used by age-1 steelhead averaged 58.5 (13.7) cm and 0.4 (0.6) cm/s, and for age-2 steelhead the mean use was 62.2 (19.4) cm and 0.4 (0.6) cm/s.

Differences were found among the microhabitats at locations used during summer by the age groups and those that were available (Fig. 2). The depths at locations used by age-0 steelhead were shallower than the depths at locations used by age-1 ($t = 7.8$;

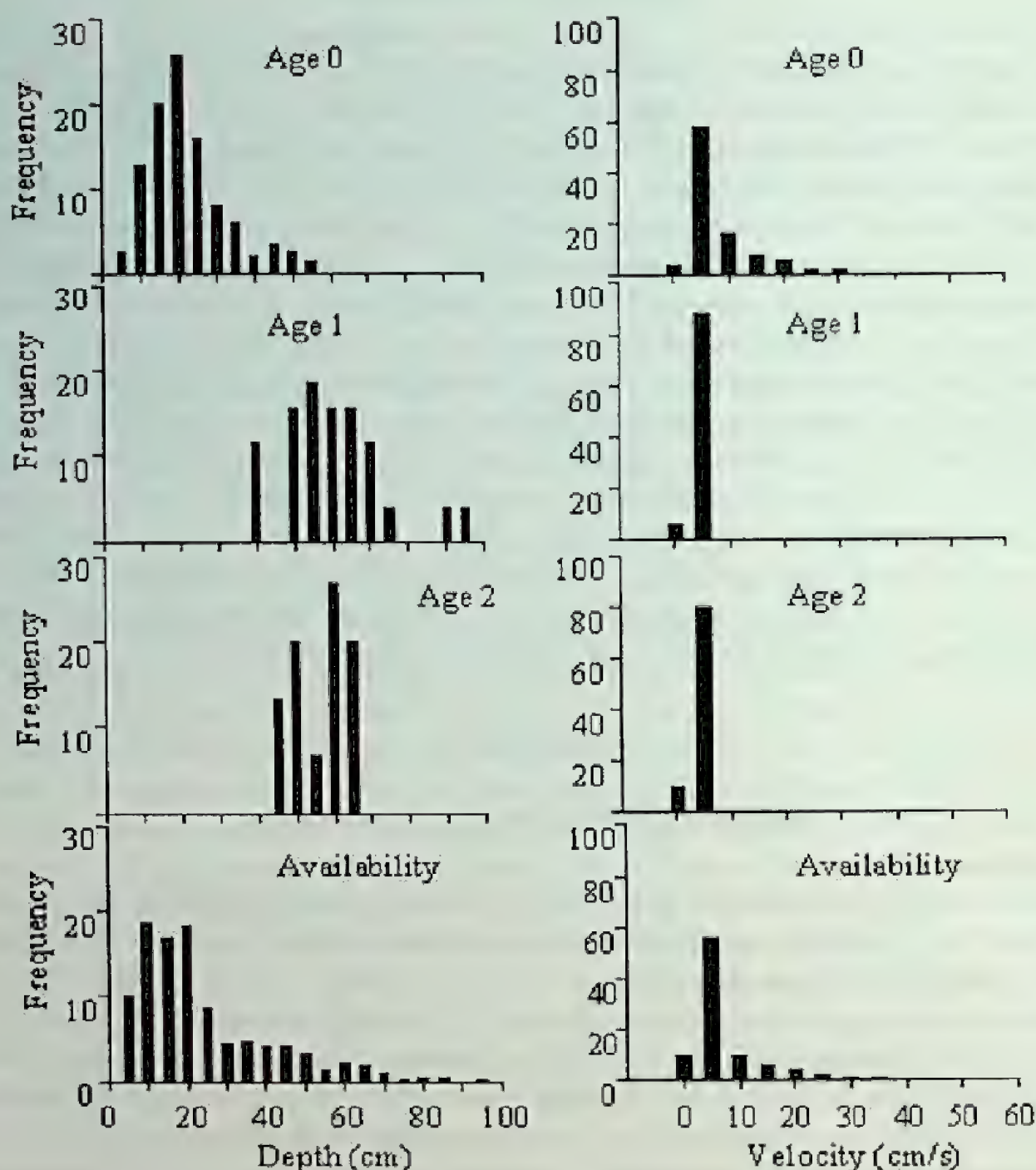


Figure 2. Relative percent frequency of occurrence of the depth and velocity microhabitats at locations used by age 0 ($n = 305$), age 1 ($n = 26$), and age 2 ($n = 15$) juvenile steelhead and those available ($n = 503$). Data collected in both spring and summer for age-0 steelhead have been combined.

$df = 10$; $P < 0.0005$) and age-2 steelhead ($t = 6.5$; $df = 9$; $P < 0.0005$), but similar to the available distribution of depths ($t = 0.3$; $df = 10$; $P = 0.8$). The depths at locations used by age-1 and age-2 fish were similar ($t = 1.2$; $df = 9$; $P = 0.3$), but deeper than what would have been used if depth had been randomly selected (both $t \geq 6.0$; $df = 10$ for age 1, and 9 for age 2; both $P < 0.0005$). Water velocities at locations used by age-0 steelhead were similar to those used by age-1 ($t = 1.4$; $df = 10$; $P = 0.2$) and age-2 steelhead ($t = 1.2$; $df = 9$; $P = 0.3$) (Fig. 2). Age-1 and age-2 steelhead used locations with similar velocities ($t = 1.1$; $df = 9$; $P = 0.3$). The distribution of available water velocities was

similar to the distribution of the velocities at locations used by age-0 ($t = 0.8$; $df = 10$; $P = 0.4$) and age-2 steelhead ($t = 1.4$; $df = 9$; $P = 0.2$). Age-1 fish used locations with velocities that were slower than what would have been used if velocity had been randomly selected ($t = 2.5$; $df = 10$; $P = 0.03$). The linear regression model indicated the depths at locations used by juvenile steelhead increased with fish length ($r^2 = 0.65$; $df = 1, 119$; $P < 0.0005$). In contrast, the water velocities at locations used decreased with increasing fish length ($r^2 = 0.46$; $df = 1, 119$; $P < 0.0005$).

Use of Run and Pool

The average (± 1 SD) density of age-0 steelhead in runs was 1.8 (0.5) fish/m², whereas in pools mean density was 1.5 (1.2) fish/m². Average density of age-1 and age-2 fish was similar between pools (age-1 fish = 0.22 ± 0.04 ; age-2 fish = 0.09 ± 0.08) and runs (0.21 ± 0.1 ; 0.08 ± 0.05). Age-0 steelhead used pools less and runs more than expected when occupancy was adjusted according to the area of the mesohabitat ($\chi^2_c = 33$; $df = 1$; $P < 0.001$) (Fig. 3). The proportional use of runs and pools by age-1 and age-2 steelhead did not differ from an expected proportion, based on mesohabitat area (Fig. 3).

Variation in the density (fish/m²) of juvenile steelhead was related to features of the sampled mesohabitats. For age-0 steelhead, not all slopes of the regression model involving area, mean depth, percent riparian canopy cover, and estimated density of age-2 steelhead were equal to zero (adjusted $R^2 = 0.89$; $P = 0.003$). Mean depth of the sampled unit did not contribute to predict the estimated density of age-0 steelhead (P

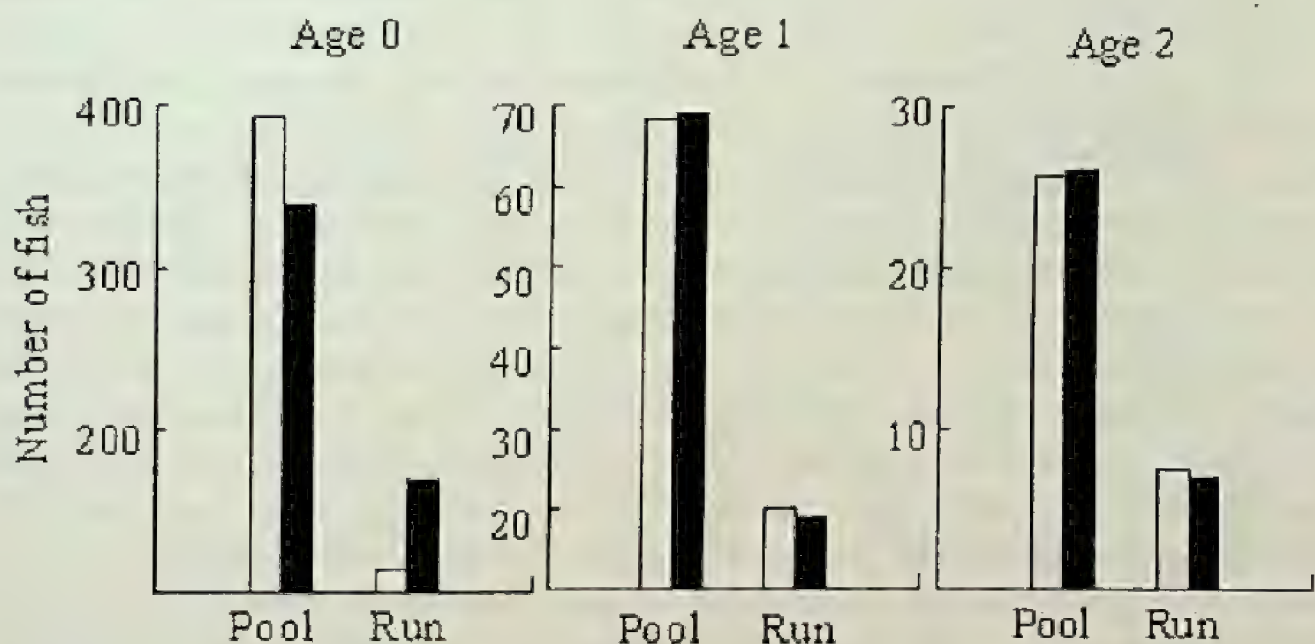


Figure 3. Expected (unfilled bar) versus observed (shaded bar) pooled estimates (number of fish) of age-0, age-1, and age-2 steelhead in pool and run determined in September 2000. Electrofishing estimates were used as the observed values and to calculate expected values, based on the area of the sampled mesohabitats.

= 0.3), but area ($P = 0.001$), amount of riparian canopy cover ($P = 0.001$) and estimated density of age-2 steelhead did ($P = 0.02$). Coefficients were positive for area and negative for estimated density of age-2 steelhead and amount of riparian canopy cover. For age-1 and older steelhead, the overall multiple regression model was significant (adjusted $R^2 = 0.96$; $P < 0.0005$). Area and mean depth of the sampled unit contributed positively to predict estimated density of age-1 and older steelhead (both $P \leq 0.01$), whereas the amount of riparian canopy cover did not contribute ($P = 0.8$).

Colonization of Pools

A total of 80 age-0 steelhead, but no age-1 or older steelhead, colonized the 12 pools that were created. Age-0 steelhead colonized deep and shallow pools similarly in terms of number of individuals per area ($t = 0.5$; $P = 0.6$) and biomass per area ($t = 1.0$; $P = 0.4$) (Fig. 4). Age-0 steelhead in shallow and deep pools were similar in terms of maximum length ($t = 0.3$; $P = 0.8$) and average weight ($t = 0.8$; $P = 0.5$). Power analyses indicated the tests had about 50 % power to detect the small mean differences between the treatment variables.

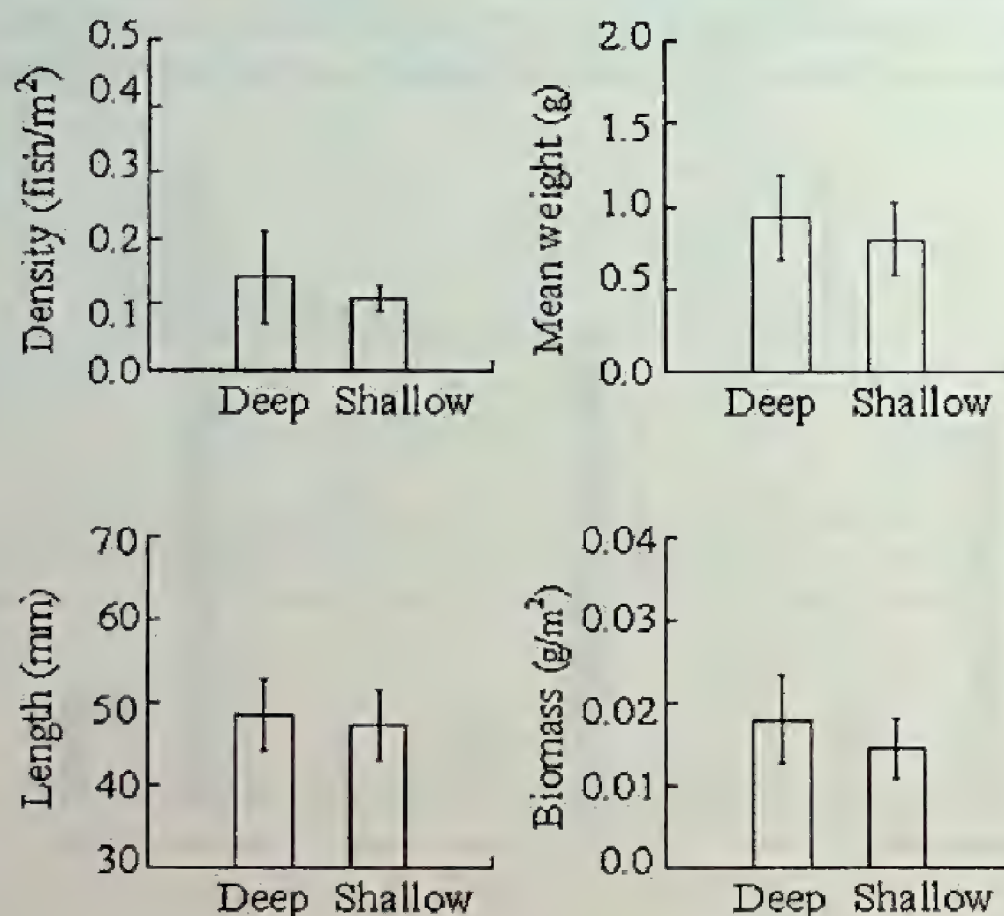


Figure 4. Density, maximum fork length, weight per individual, and biomass of age-0 steelhead that colonized the created shallow (maximum depth: 12-16 cm) and deep (30-40 cm) pools after 5 weeks during summer. Top of bar is the mean (± 1 SE). For each treatment type $n = 6$.

DISCUSSION

Habitat Use Patterns

The patterns of habitat use determined for age-0 steelhead in this study are consistent with those of other investigators reporting on habitat use in age-0 salmonids. Age-0 salmonids use locations in streams that are shallow (Shapovalov and Taft 1954, Hillman et al. 1987, Bisson et al. 1988, Moore and Gregory 1988), and can occupy shallower water than larger, older conspecifics (Symons and Heland 1978, Rosenfeld et al. 2000, Spina 2000, Muhlfeld et al. 2001). The ranges of water depth and velocity used by age-0 steelhead in this study were similar to those reported for age-0 fish in northern territories (Everest and Chapman 1972, Johnson and Kucera 1985, Sheppard and Johnson 1985) even though there were likely differences in habitat availability among study areas. Because the scope of this study was limited in time and space, additional research is needed to fully evaluate the similarity of habitat associations documented for steelhead in Santa Rosa Creek and those expressed by steelhead in other streams.

Juvenile steelhead showed an ontogenetic shift in use of microhabitat. Shifts in use of depth with increasing size or age of salmonids are common (Shapovalov and Taft 1954, Baltz et al. 1991, Allen 2000, Rosenfeld et al. 2000), and in this study water depth was a good predictor of fish length. Although juvenile salmonids reportedly use deeper and faster water as they grow (Everest and Chapman 1972, Hillman et al. 1987, Muhlfeld et al. 2001), at Santa Rosa Creek water velocities at locations used did not increase with fish length, but rather smaller fish used locations characterized by water velocities that were faster than those used by larger conspecifics. The finding that steelhead in Santa Rosa Creek did not use faster water as they grew probably reflects the extremely low discharge and the limited range of water velocities, a common dry-season feature of unregulated streams near the southern extent of the species' range.

Generally, juvenile steelhead use mesohabitat types exhibiting moderate to swiftly flowing water (Shapovalov and Taft 1954, Bisson et al. 1988, Roper et al. 1994); in this study all age groups used pools and runs, but age-0 steelhead showed greater occupancy of runs than was expected. Greater occupancy of a particular mesohabitat type does not necessarily indicate the mesohabitat type itself is preferred. Rather, the greater occupancy can be an artifact of a response to biotic conditions or the preference (or requirement) for an innate mesohabitat property (e.g., water depth or velocity, substrate). This latter point is suggested by the findings of other investigators, which show the mesohabitat type occupied by age-0 and age-1 steelhead changed across reaches within a single stream (Roper et al. 1994). Thus, the greater occupancy of runs by age-0 steelhead at Santa Rosa Creek could be the result of a preference for shallow water, which age-0 steelhead used instead of deep water. Because juvenile steelhead at Santa Rosa Creek shifted use of microhabitat to deeper water as they grew, larger individuals would be expected to show greater occupancy of deep-water mesohabitat types such as pools (Bisson et al. 1988, Heggnes et al. 1991, Rosenfeld et al. 2000, Muhlfeld et al. 2001). This was not the case for age-1 and age-2 steelhead in Santa Rosa

Creek when occupancy was adjusted to the size of the sampled mesohabitat. Habitat occupancy by a species can expand or contract among habitats over time in response to fluctuations in the abundance of the overall population (e.g., Fretwell and Lucas 1970, Fretwell 1972, Fraser and Sise 1980) and, therefore, the findings should be interpreted carefully.

A size-selective predation risk and intraspecific interactions could have contributed to the bigger-fish deeper-water pattern of microhabitat use noted in this study. The fact that age-1 and older steelhead showed a preference for deep water suggests importance of deep water to these age groups, possibly for avoiding predators that select large fish in shallow water (Power et al. 1989, Harvey and Stewart 1991). By contrast, older larger salmonids can seek deep water because growth of the larger fish can be greater in deep water than in shallow water (Rosenfeld and Boss 2001), and interactions with smaller, younger steelhead in shallow water can inhibit growth of the older steelhead (Harvey and Nakamoto 1997). The use of shallow water by age-0 fish can be a response to older conspecifics in deep or preferred areas of the creek (Bohlin 1977, Symons and Heland 1978, Freeman and Stouder 1989). Density of age-0 steelhead was negatively related to density of age-2 steelhead, and based on my underwater observations and the findings of this study, age-0 steelhead appeared to avoid deep areas of the creek, which were occupied by larger conspecifics. The negative relationship between the density of age-0 and age-2 steelhead could be an artifact of a size-specific microhabitat requirement rather than due to age-2 steelhead, however.

The effect of the deep and shallow treatments on colonization by age-0 steelhead was neither statistically nor biologically significant (i.e., extremely small mean differences between treatment variables). The conspicuous absence of age-1 and older fish from the created pools is interesting, but whether the lack of settlers is related to the depth of the created pools or some other inherent factor (such as the physical complexity of the pools) is unknown. Although mean depth was similar between created (i.e., deep treatment) and natural pools, maximum depth was not. Therefore the created pools were probably not sufficiently deep for age-1 and older steelhead.

Besides depth, there are other factors that contribute to define the appropriateness of mesohabitat for older conspecifics. One potential factor is physical complexity. Natural pools typically possessed root wads, cobble-boulder accumulations, and undercut banks, and age-1 and older steelhead selected these elements for shelter, based on my observations. By contrast, shelter was absent from the created pools. Thus, the plainness of the created pools could have contributed to the lack of age-1 and older fish settling in these pools. The extent that a lack of movement or a cost of exploration contributed to the absence of age-1 and older steelhead in the created pools is unknown, but deserves consideration. I assumed all age groups engaged in some level of movement during the experiment because among other reasons the transition between mesohabitats (with the exception of the few riffles I observed) in Santa Rosa Creek appeared to be adequate for movement. I did not evaluate this assumption, however, and therefore whether age-1 and older steelhead were truly encountering the created pools is unclear. Overall, the reason for the lack of older settlers in the created pools warrants further investigation.

The age groups showed microhabitat use patterns that contribute to maximizing use of living space, thereby decreasing intraspecific niche overlap and encouraging coexistence of conspecifics (Van Horne 1982, Sandheinrich and Hubert 1984, Kincaid and Cameron 1985). In Santa Rosa Creek, the size-specific use of depth caused the age groups of steelhead to segregate along depth gradients and different areas within mesohabitat types, in particular pools (cf. Spina 2000), based on my underwater observations. Partitioning of living space may increase the chance of survival and multiyear freshwater residence of juvenile steelhead especially during the summer and late fall when the quantity of surface water, and therefore living space, typically decreases, potentially increasing the likelihood of intercohort interactions. There might be times when microhabitat partitioning becomes critically necessary; all age groups may congregate in pools during periods of elevated water temperature (Nielsen et al. 1994, Matthews and Berg 1997) and perhaps during seasonally reduced surface flow, prevalent conditions near the southern extent of the species' range. Nevertheless, the potential for competitive interactions was not entirely precluded because considerable overlap in use of depth remained between age-1 and age-2 steelhead, despite the size-specific use of depth.

The findings obtained from the multiple linear regression analysis showed that density of age-0 steelhead increased as the percentage of riparian canopy cover decreased. Reaches with open canopy can increase primary production and abundance of young salmonids (Hetrick et al. 1998a, b). The finding that indicates mesohabitats with less riparian canopy produce more age-0 steelhead should be interpreted carefully because streamside vegetation provides numerous functional values to stream-dwelling fish. These values include streamside and instream cover (Bustard and Narver 1975, Wesche et al. 1987), a source of woody debris and therefore instream habitat for fish (review by Bryant 1983, Lisle 1986, Shirvell 1990), a filter for sediment (Cooper et al. 1987), and temperature regulation and streambank stability (review by Platts 1991). In Santa Rosa Creek, accumulations of leaf litter at the tail of pools appear to increase residual pool depth during low-flow conditions, and leaves along the creek bed are used by age-0 steelhead for cover (A. Spina, personal observations). Multiyear freshwater residence of juvenile steelhead requires habitat that is appropriate for all age classes, not simply age-0 steelhead; the results of this study suggest that habitat used by age-0 steelhead might not be suitable for age-1 and older steelhead.

Management Implications

Based on the value of shallow and deep water and constituent mesohabitats to juvenile steelhead, and the fact that some stream alterations can negatively impact salmonids (review by Hicks et al. 1991), activities that encroach into shallow water or cause loss of deep water are likely to adversely affect this species. Stabilizing stream banks or road slopes or constructing bridge abutments in a manner that intrudes into shallow water may reduce the amount and suitability of rearing areas for age-0 steelhead, given the preference of shallow water by this specific age group in Santa Rosa Creek. Moreover, changes in discharge that alter the amount of shallow water

may reduce the appropriateness of rearing areas for age-0 steelhead as well. Based on the findings that indicate age-1 and older steelhead preferred deep water, activities causing a loss of deep water might reduce abundance of the larger older age groups. Many of the pools in Santa Rosa Creek were formed by scour associated with downed trees or root wads, and age-1 and older steelhead were abundant in pools (cf. Fausch and Northcote 1992, Rosenfeld et al. 2000). Activities that cause a decrease in the amount or size of woody debris in streams can reduce the amount of pool habitat (Montgomery et al. 1995) and abundance of salmonids (Elliott 1986, Fausch and Northcote 1992). Although avoiding alteration of the stream environment is desirable particularly where listed species are of concern, such intentions seem unrealistic where urban areas or existing developments overlap with the species. Knowledge of habitat associations in steelhead trout is therefore useful for predicting the potential effects of planned actions on steelhead and their habitat, and as a basis to modify such actions for minimizing the adverse effects on this species. Ideally, site-specific or region-specific data are needed for evaluating the likelihood and identifying the type of adverse effects that could result from proposed actions, identifying appropriate protection measures to be incorporated into the proposed actions, and ultimately developing proposed actions that have minimal adverse effects on this species. The results of this study provide information concerning habitat associations of juvenile steelhead near the southern extent of their range, and therefore can be used in combination with other existing information to support the work of biologists for protecting this species and guiding recovery efforts.

CONCLUSION

The habitat associations documented for juvenile steelhead in a small south-central California stream appear to be influenced by water depth and to a lesser extent water velocity, and are believed to reflect size-specific microhabitat requirements and biotic interactions. Variation in the density of juvenile steelhead was related to the water depth and size of the sampled units and the amount of riparian canopy cover. Because the habitats used by age-0 steelhead might not be suitable for age-1 and older steelhead, the results suggest that multiyear freshwater residence of steelhead requires habitat that is appropriate for all age classes. The results represent only a narrow view of habitat use by steelhead; more research on additional streams at and near the southern extent of the species' range is needed to fully assess the range and similarity of habitat associations expressed by southern steelhead over time and space.

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A TEMPORAL, SEX-SPECIFIC OCCURRENCE PATTERN AMONG WHITE SHARKS AT THE SOUTH FARALLON ISLANDS, CALIFORNIA

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Based on 239 observations of 22 known-individual white sharks from 1987 to 2000, we report a temporal, sex-specific occurrence pattern among adult white sharks at the South Farallon Islands (SFI), California: individual males may occur every year, whereas individual females show a biennial occurrence pattern, being recorded every other year at most. This sex-specific occurrence pattern implies a 2-year reproductive cycle, resulting in a lower reproductive potential than previously thought, which has important implications for the conservation of this species. These results also suggest that female white sharks may travel significant distances in the North Pacific Ocean during a biennial reproductive cycle to give birth, whereas copulation may occur closer to northern California, allowing males to return annually to SFI.

INTRODUCTION

Segregation by sex has been reported for several species of sharks (Bigelow and Schroeder 1948, Bullis 1967, Pratt 1979, Gilmore 1993). Skewed sex ratios among capture data have indicated sex-specific aggregation patterns according to both geographic area and water depth. In the white shark (*Carcharodon carcharias*), spatiotemporal segregation by sex has been reported in North America (Casey and Pratt 1985, Klimley 1985), South Africa (Ferreira and Ferreira 1996), and southern Australia (Bruce 1992, Strong et al. 1996). Off the Pacific coast of North America, capture records suggest that higher proportions of adult (mature) male white sharks occur to the north whereas more juveniles and adult females have been recorded to the south, although this sex-specific pattern was not statistically significant (Klimley 1985). Speculation on sex-specific segregation in the white and other sharks has focused on sex-specific roles in reproduction (Gilmore 1993); however, annual reproductive behavior patterns at the population level are poorly understood.

White sharks occur at the South Farallon Islands (SFI), located 48 km off San Francisco, California, during autumn (primarily September-November) to prey on pinnipeds but are absent there during spring (Ainley et al. 1985, Klimley et al. 1992, Pyle et al. 1996). Although both male and female white sharks have been recorded inter-seasonally at SFI (Klimley and Anderson 1996), it is unknown whether or not sex-specific segregation patterns occur. Here we report a temporal, sex-specific occurrence pattern among adult white sharks at SFI based on examination of inter-seasonal return patterns of known-sex individuals.

METHODS

From 1987 through 2000, during daily observations in autumn (1 September to 30 November), we identified and documented individual white sharks using size and unique markings such as scars, mutilated fins, natural pigmentation patterns (which remain static from year to year; pers. obs.), and the distribution of notches on the trailing edge of the dorsal fin. From 1987 to 1992, sharks were documented with still photographs (Anderson and Goldman 1996) and shore-based video recorders (Klimley and Anderson 1996). By 1993 we discovered that white sharks investigated small (< 6 m) vessels or decoys (Anderson et al. 1996), particularly during and up to 2 hours subsequent to feeding events on pinnipeds. This behavior allowed us to employ underwater video recorders mounted on poles to document individual sharks and confirm sex by the presence (male) or absence (female) of claspers (Pratt 1996). The absence of claspers was sometimes difficult to confirm; thus, sharks were only sexed as female using adequate video-documentation of the shark's ventral region. The analysis presented here includes all individual adult sharks that 1) had distinctive features allowing confirmed identification, 2) were observed in 2 or more years, and 3) were of known sex.

RESULTS

During the 14-yr study period, we made 239 observations of 22 distinctively marked white sharks (8 females and 14 males) recorded during 2 or more years (Table 1). These sharks ranged from 3.2 to 5.9 m total length, as estimated by comparison with our 4.2-m and 5.2-m research vessels. Of these observations, 82 (34.3%) were of females and 157 (65.7%) were of males, a 1:1.91 ratio. When inter-seasonal occurrence patterns are examined (Table 1), all 8 females were observed during odd-numbered years or even-numbered years but not both, and all 14 males were observed in both odd- and even-numbered years. This sex-specific difference was highly significant (Pearson's $\chi^2 = 22.0$, $P < 0.0001$); thus, male white sharks appear to have an annual occurrence pattern at SFI in contrast to the biennial pattern of females. Our observed sex ratio of 1:1.91 (females to males) can therefore be explained by this sex-specific pattern occurring within a population of balanced sex ratio: each year all of the males but only half of the females of the SFI population are present.

DISCUSSION

Previous reports that female white sharks occurred in consecutive years at SFI (Klimley and Anderson 1996) were likely based on misidentified or mis-sexed sharks from land-based video recordings and visual observation only (A.P. Klimley, pers. comm.). For example, shark "CC", labeled female in Klimley and Anderson, is male 8801 of this paper (claspers repeatedly confirmed with underwater video recordings), and the observation in 1992 of the female shark "AC" of Klimley and Anderson (8901 of this paper) was very likely based on a mis-identification of the male 9601. Sharks 8901 and 9601 had very similar mutations to the caudal fin and 9601 was frequently recorded in

Table 1. Number of observations each year of individual white sharks at the South Farallon Islands, California.¹

Shark	Year													
	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
A. Females														
8901			2		6		7		1					
8902			1		1		5							
8903			1		1		1							
8904			2		1		3		5					
9201						10				11		2		4
9203						4		2		3				
9401								2		2				
9801												2		3
B. Males														
8701	1				1		1			3		1		
8801		1		1		1	2	2		6		4	1	1
8802		1		3	15	16		2						
9001				1		1	1							
9101					3			2		4				
9202						3		3	4	2		3	1	
9301							1	5	7	1		1	5	4
9302							4	1		3		1		
9306							1	4		5				
9502									3			2		
9601										3	1			
9602										3	1		1	1
9901													2	2
9902													3	1

¹An "observation" is defined as at least one sighting of an individual on a particular date.

the location of the 1992 sighting (see Klimley and Anderson 1996), whereas 8901 was never recorded in that vicinity and would be unexpected there (see Goldman and Anderson 1999). Repeated documentation of many individuals between 1993 and 2000 with video-recorders indicated that no females were present in consecutive years (Table 1).

The gestation period and length of reproductive cycle in mature female white sharks remains unknown. Studies of pregnant females off Australia and New Zealand suggest that both copulation and parturition occur there in the austral spring and summer (Francis 1996). Francis speculated that copulation might occur immediately subsequent to parturition, and that females might be able to carry successive litters with little or no resting period, but acknowledged that this needed to be demonstrated. Alternatively, the biennial occurrence of adult female white sharks at SFI suggests a 2-year reproductive cycle, during which either gestation is prolonged (> 1 yr) or copulation occurs well after parturition. Mollett et al. (2000) summarize evidence suggesting prolonged gestation periods (up to 18-24 months) and 2-year or 3-year reproductive cycles in several species of large sharks. Their speculation of an 18-month gestation for the white shark would be consistent with the 2-year reproductive cycle indicated by our data. A biennial reproductive cycle would result in a lower reproductive potential in this species than previously thought (litter sizes of only 4-11 pups have been reported; Francis 1996), which has important implications for the conservation of this species, particularly from the effects of incidental mortality (Wintner and Cliff 1999).

Based on capture data, Klimley (1985) suggested that both female and male white sharks moved northward from southern California waters into northern California waters as they matured (to 3-4 m length) and began to feed on marine mammals. Females appeared subsequently to return to southern California waters to pup whereas males remained in northern California waters. Although our data are not inconsistent with this hypothesis, recent evidence (Boustany et al. 2002, unpubl. data) indicates that both male and female white sharks can travel from SFI into the central Pacific Ocean during the spring and summer. Pregnant females may be acquiring and storing energy during the autumn at SFI for parturition and copulation the following spring and summer, after which it may take them a full year to return to northern California to gain energy for a subsequent reproductive attempt. Alternatively, females could travel from SFI to the central Pacific well off Baja California (see Boustany et al. 2002) for copulation, after which they may travel to an unknown location for parturition before returning to SFI 18-20 months later. In either case, female white sharks could potentially travel significant distances in the North Pacific Ocean during a biennial reproductive cycle to give birth, whereas copulation may occur closer to northern California, allowing males to return annually to SFI. Pregnant females have been captured in the western Pacific ocean (Bruce 1992, Francis 1996, Uchida et al. 1996) but are thus far unknown from the eastern North Pacific.

Recent genetic evidence suggesting that female white sharks in the Southern Hemisphere are philopatric whereas males travel widely (Pardini et al., 2001) appears to differ from our results. Based on our observational data, we suggest either that the genetic evidence indicates long-term, natal-dispersal patterns at the population level

(as opposed to the short-term migration patterns at the individual level reported here), or that differing migration or reproductive strategies occur in different populations of the species. We are currently deploying long-term (9-12 month) satellite transmitters on female and male white sharks at SFI to better understand sex-specific occurrence patterns in the North Pacific.

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RESPONSES OF SAN JOAQUIN KIT FOXES TO AN OIL-GAS WELL FIRE

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Well blowouts are rare, but they have the potential to affect local wildlife in oil producing areas (Odu 1972, Freedman 1995). A blowout is defined as the uncontrolled flow of natural gas, petroleum, or water from a well bore (Freedman 1995). Well blowout impacts on upland species have been reported before (Warrick et al. 1997), but more information is necessary to further understand the potential effects of oil-gas extraction on wildlife.

Here I describe the movements of four endangered San Joaquin kit foxes, *Vulpes macrotis mutica*, before and after a well blowout. Other risks in oilfield environments include the potential presence of hydrogen sulfide gas in kit fox dens (Spiegel and Dao 1997), an increase in concentrations of Zn, Cu, and Cd in kit fox tissues (Charlton et al. 2001), and oilfield-related mortalities such as exposure to toxic chemicals (e.g., arsenic) and kit foxes being covered in oil (Cypher et al. 2000). The San Joaquin kit fox was listed as endangered by the U.S. Department of the Interior in 1967 and as threatened by the State of California in 1971 (U.S. Fish and Wildlife Service 1998¹).

Well blowouts and fires have occurred in San Joaquin kit fox habitat before: 1994 well blowout on Naval Petroleum Reserve No. 1 (Warrick et al. 1997); Elk Hills well fire of 1977 (R. Hauser, Department of Conservation, Division of Oil, Gas, and Geothermal Resources, pers. comm.); however, no information was gathered on kit fox behavior. There have been 16 gas or oil blowouts since 1990 in Kern County, California; they lasted a few hours to a few days and did not result in fires (R. Hauser, pers. comm.), too short for a study on blowout effects on kit foxes.

The Bellevue East Lost Hills No. 1 (17-26S-21E) oil-gas well fire provided an opportunity to observe kit fox reaction to the incident. The oil-gas well was located ~300 m east of the California Aqueduct (aqueduct), in a 1200-ha cotton field near Lost Hills, Kern County, California (35.6°N, 119.7°W). The well blew out at approximately 20:00 hrs on 23 November 1998. It burned for 15 days, blew steam and minute oil droplets for 6 days, was intentionally re-lit, and then burned for another 6 months. Potential disturbance during the entire 6-month period included a roaring noise from escaping gas, intense heat and light from the fire, smoke and toxic fumes, ground vibration, and increased vehicular and human activity.

Since May 1997, San Joaquin kit foxes have been radio-collared and monitored from one to four times a week along a 24-km stretch of the aqueduct between Lerdo Highway

¹U.S. Fish and Wildlife Service. 1998. Recovery plan for upland species of the San Joaquin Valley, California. Region 1, Portland, Oregon. 319 pp.

north to the Kern County line for an agricultural land use study. Foxes were radio-tracked using two truck-mounted null tracking systems with paired 2-element antennae (Kenward 1987). Stations were located along access roads of the aqueduct and separated by approximately 800 m. After setup and calibration of the systems, researchers at two adjacent stations simultaneously took bearings on radio-collared kit foxes (Mech 1983).

A telemetry session was initiated approximately 1 h before sunset and continued for approximately 4 h. The first 3-5 h after sunset is typically when kit and swift fox activity is highest (Cypher et al. 2000). Locations were collected on all collared foxes in the vicinity and successive locations on individual foxes were separated by ~10 min. Home range data on these foxes were collected from May 1997 until the fox experienced mortality. Behavioral observations reported here were made during November 1998 (pre-blowout) and in December 1998 (post-blowout).

Before the blowout, two radio-collared kit foxes were active near the well and two were not. All four kit foxes had linear shaped home ranges, closely associated with the aqueduct corridor based on 1.5 yrs of telemetric locations. The foxes normally used the aqueduct right-of-way and an earthen berm 4 m in height (paralleling the aqueduct) as a travel corridor within their home ranges. Foraging in the adjacent agricultural fields occurred frequently but was not observed at distances >700 m from the aqueduct.

An adult male kit fox (M1) used a home range 10 km south of the oil-gas well site. On the evening of the well explosion (23 November 1998), he traveled along the aqueduct right-of-way to an area 3.5 km north of the well site. This was the first time he was found in that area. After November 23, he stayed in this area, with his home range overlapping a female kit fox (F1) in the study area. She used an area extending from 5 to 10 km north of the well site. She remained in this home range both before and after the well fire.

Prior to the blowout an adult female kit fox (F2) had a home range that extended from the well to 5 km north of the well site. After the fire, her southern home range boundary extended south another 2 km and she continued to use the northern portion of her range. She was found dead 1.6 km south of the well site along the aqueduct on 7 December 1998. This fox was killed by a red fox, *V. vulpes*, but was not eaten.

The fourth kit fox, an adult male (M2), also ranged in this area. Prior to the well fire, his home range overlapped the home ranges of the two females, F1 and F2 and extended from 1 km south to 6 km north of the well site. After the fire, he abandoned his northern range area, and remained in the southern portion of his range, shifting an additional 1 km to the south. He was found 300 m to the west of the well on two occasions and 600 m northwest of the site on one occasion. He was killed by a coyote, *Canis latrans*, and found dead on 23 December 1998, 13 km north of the well.

Although there were too few data to quantitatively analyze during the blowout period, it appears from visual and telemetric observations that the impact of the gas-well fire on these four kit foxes was minimal. Kit foxes F1 and M1 were the least affected. F1 maintained the same home range throughout the incident, and pair-bonded with M1 upon his arrival to her range area. M1 moved through the area of the well fire most likely by using the 4-m berm as a corridor and heat-noise barrier. Male kit foxes usually join

the females at natal dens in October or November and breeding occurs during December or January (McGrew 1979). M1 and F1 had a litter of three male kits in February 1999, consistent with the hypothesis that M1's movements were of a pair-bonding nature, and not necessarily a reaction to the well fire.

Kit fox F2 shifted her range to the south of the well site. M2 may have formed a pair bond with F2 because he shifted his range south as well, overlapping her new area. However, F2 was attacked and killed by a red fox. Subsequently, M2 moved 13 km to the north into unknown territory and was killed by a coyote.

Although an oil-gas well burned nearby, the female foxes continued with their behavioral activities and did not move out of their range areas. The male foxes did have large movements, but it is believed they are related to breeding season behaviors (McGrew 1979, Zoellick et al. 1989, 2002), rather than a well fire reaction.

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